The Effect of Open Marsh Water Management Practices on the Carbon Balance of Tidal Marshes in Barnegat Bay, New Jersey

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Abstract

The Effect of Open Marsh Water Management Practices on the Carbon Balance of Tidal Marshes in Barnegat Bay, New Jersey
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Salt marshes have been modified for mosquito control since the early 1900s. Open marsh water management (OMWM) is a mosquito control technique first applied in New Jersey in the 1950s. It has been extensively used in the mid-Atlantic and also in Massachusetts, New York, Connecticut, Florida, and Louisiana. This practice involves excavation of areas of marsh, constructing shallow ponds to provide habitat for mosquito larvae-eating fish with the goal of reducing mosquito populations. Nearly half of Barnegat Bay, New Jersey's coastal marshes have been subject to OMWM since 1976, and its effects on ecosystem function are not well understood. Here, we assess the impacts of OMWM on the carbon balance of salt marshes associated with the conversion of intact vegetation to open water ponds. Using a precise object-based image analysis classification technique, we found that over 7,000 ponds, approximately 3% of Barnegat Bay's coastal marshes, have been constructed for mosquito control. Using state of the art IR technology to examine in-situ greenhouse gas fluxes in Barnegat Bay, we found evidence to suggest that the conversion from intact plants to open water ponds negatively shifted the carbon balance of the salt marsh, both within the footprint of the constructed pond as well as within the area where deposition of sediment has prevented the recolonization of marsh vegetation. We conclude that this management technique has significantly reduced the carbon sequestration capacity of Barnegat Bay's tidal wetlands.

1. Introduction

Coastal wetlands are the transition zone between marine and terrestrial ecosystems (Rochlin et al., 2012). Coastal wetlands provide crucial ecosystem services, which include acting as fish nurseries, improving coastal water quality through the sequestration and removal of sediment and nutrients, providing shoreline protection through wave attenuation and storm surge reduction, and sequestering globally significant volumes of organic carbon (Rochin et al., 2012). These coastal ecosystems store carbon as aboveground biomass (i.e. leaves, stems, branches), as belowground biomass (i.e. roots, rhizomes), and within carbon-dense organic soils, where anoxic conditions prevent or slow the remineralization of organic matter. Salt marshes additionally trap allochthonous sediment via emergent stems and accrete vertically, thus, they do not have a carbon saturation point (Chmura et al., 2003; Mcleod et al., 2011). Because of this vertical accretion capacity, carbon burial by vegetated coastal ecosystems has been estimated at 100 teragrams of carbon per year (Mcleod et al., 2011; Hopkinson et al., 2012), corresponding with an organic carbon sequestration value of ~ 20 to 2000 grams of carbon per square meter per year (Mcleod et al., 2011).

Recently, intense scientific attention has been focused on the carbon sequestration capacity of vegetated coastal ecosystems (termed ‘blue carbon’). This is because the conservation and restoration of such environments can provide environmental benefits through emissions mitigation as well as other ecosystem services valuable to coastal communities facing climate change. In addition, carbon finance is being used to subsidize restoration projects, expanding and diversifying the sources of funding available for coastal conservation and restoration (Galatowitsch 2009). However, both climate change

and anthropogenic disturbances threaten the continued survival of vegetated coastal ecosystems.

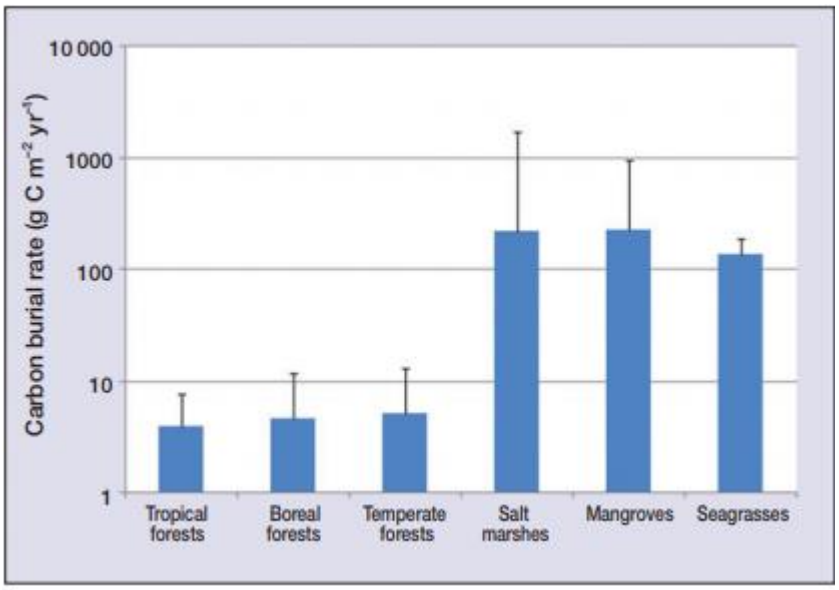

Figure 1. Average long-term rates of carbon sequestration in terrestrial forest soils and vegetated coastal sediments from *Mcleod et al., 2011*.

Alteration of U.S. coastal marshes began with European colonization (Sebold, 1992; James-Pirri et al., 2012), and initially included alterations for grazing, agriculture, salt hay farming, and waterfowl impoundments (Sebold, 1992). Salt marshes provide a suitable breeding habitat for salt marsh mosquito species (Rochlin et al., 2012), and have been physically modified to control mosquito populations since the early 1900s (Lathrop et al., 2000). These mosquitos are of public health concern, as they are vectors for Eastern equine encephalitis and West Nile virus. Efforts to control mosquitos via drainage led to installation of parallel grid ditching in almost 95% of the tidal marshes between Maine and Virginia, most during the Works Progress Administration in the 1930s (Lathrop et al., 2000; Rochlin et al., 2012; Quirk et al., 2015). Alterations of the salt marsh caused changes to the physical and chemical soil environment over short time

scales (Quirk et al., 2015). Ditching lowered water tables, reduced sedimentation, and increased soil aeration, decomposition, and compaction (Quirk et al., 2015). Water table dynamics and topography in the marsh setting influence surface sedimentation, soil development, porewater chemistry, vegetation community, and plant productivity; therefore, altering these systems has had strong impacts to ecosystem structure and function (Quirk et al., 2015). Because of concern with the negative impacts to wildlife, such as fish and water fowl which utilized ponded areas, grid ditching for mosquito control was phased out by the 1960s (Lathrop et al, 2000; James-Pirri et al., 2012).

In an attempt to minimize the use of pesticides and the reduce the negative impacts on hydrology and vegetation of salt marshes from ditching, an alternative practice to control mosquitos was developed in New Jersey in the 1960s (Ferringno et al., 1968; Lathrop et al, 2000; James-Pirri et al., 2012). Termed open marsh water management (or OMWM); the objective of OMWM is to reduce mosquito breeding from a given area of marsh while enhancing wildlife habitat (Ferringno et al., 1968). This practice has been used extensively in the mid-Atlantic, Massachusetts, Connecticut, New York, Florida, and Louisiana (Quirk et al., 2015). The techniques used in OMWM are regionally dependent. In New England, ponds have been created through ditch plugging while in the Mid-Atlantic region, radial ditches and isolated ponds have been constructed (James-Pirri et al., 2009; Quirk et al., 2015). Ditch plugging involves the plugging the seaward end of present ditches to create pools (Vincent et al., 2012). Radial ditching involves the connection of present ditches to isolated ponds (Ferringno et al., 1968; James-Pirri et al., 2009). These plugs and radial ditches connect to the historic ditches, allowing high tide to bring mosquito larvae-eating fish to the ponds to reduce the larval

population in a given area (Ferringno et al., 1968; Quirk et al., 2015). In this study, we focused on isolated ponds (Figure 2).

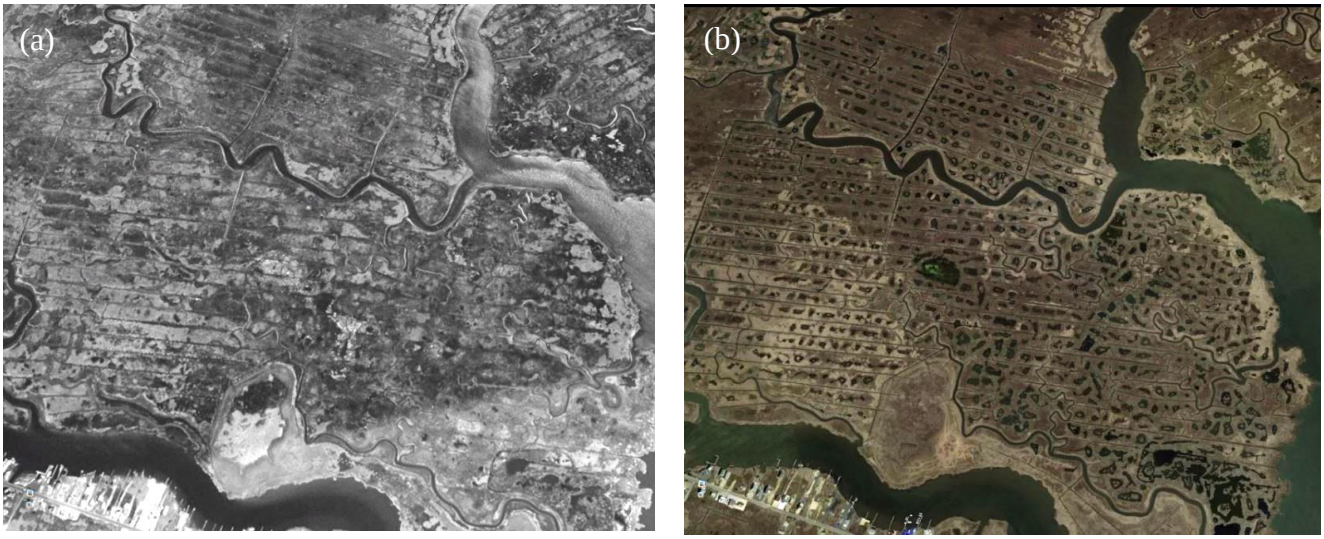

Figure 2. Example of the OMWM manipulations performed in Barnegat Bay, NJ: (a) before (1995) and (b) after isolated pond OMWM manipulation (2016).

While changes in carbon storage resulting from OMWM pond construction in wetlands have not previously been reported (James-Pirri et al., 2012), we expect that pond construction could reduce wetland carbon sequestration through two mechanisms. First, we expected that conversion of vegetated wetlands (a greenhouse gas sink) to open pond (a greenhouse gas source; Moseman-Valtierra 2016) would decrease the overall carbon sequestration of the landscape. Conversion from vegetated plants to open water would presumably decrease the carbon sequestration of the area. Since the OMWM ponds do not have macrophytes, we would not expect them to contribute to photosynthesis and ecosystem respiration (Figure 3). Second, we expected that carbon previously sequestered in vegetated wetlands that is sidecast during pond excavation will be remineralized. This mechanism is driven by oxygen-dependent remineralization processes. When salt marsh peat is anoxic, organic matter is remineralized to the environment only via less efficient mechanisms (e.g., coupled with sulfate or nitrate

reduction; Figure 3). When peat is exposed to a well-oxygenated environment it can be remineralized rapidly. Consequently, we expect marsh peat sidecast as part of pond construction would lose carbon due to decomposition and remineralization of organic matter.

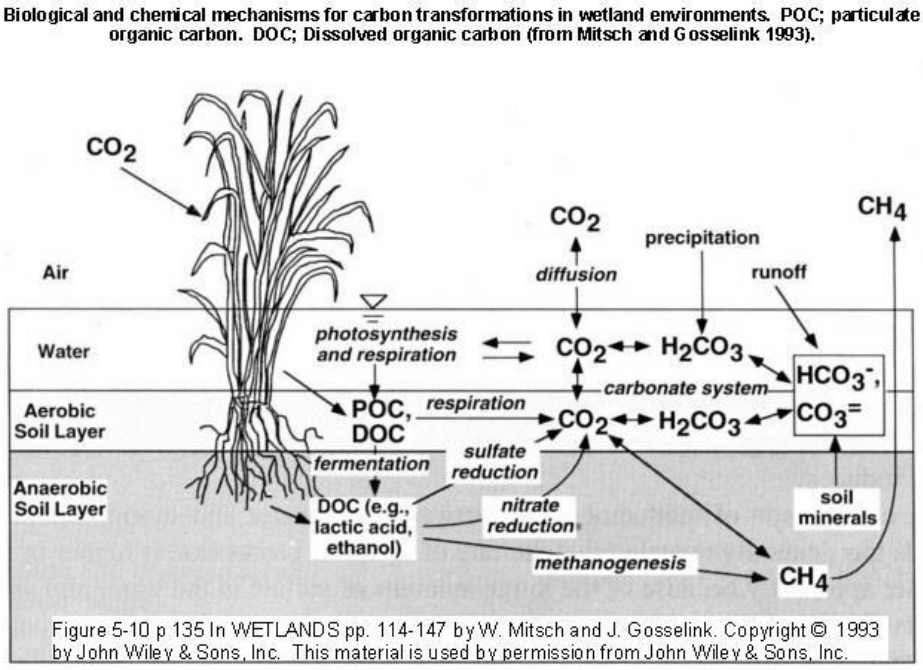

Figure 3. The carbon cycle seen in tidal wetlands by Mitsch and Gosselink et al. (1993). The major processes include photosynthesis, ecosystem respiration, fermentation, methanogenesis, and methane oxidation.

In this study, we focus on the effects of OMWM management to carbon sequestration due to the conversion of intact vegetation to open water ponds. The objective of our study was two-fold: to enumerate the area of marsh that was disturbed by pond excavation for mosquito control; and secondly to compare greenhouse gas exchange among habitats to estimate the effects of this disturbance on carbon sequestration and greenhouse gas exchange. To quantify the areal extent of habitat conversion, we enumerated the number and extent of ponds constructed for mosquito control using object-based image analysis of high resolution multi-spectral aerial imagery. To quantify changes in carbon sequestration, we measured the exchange of carbon dioxide

and methane in marsh, ponded areas, and bare ground at three different marshes across the salinity gradient of Barnegat Bay over the course of a year. Many thousands of ponds have been constructed in Barnegat Bay for mosquito control: our goal was to examine the effects of this disturbance on wetland carbon sequestration.

2. Methods

2.1 Study Site

The Barnegat Bay-Little Egg Harbor estuarine system consists of three shallow coastal bays: Barnegat Bay, Manahawkin Bay, and Little Egg Harbor (Kennish, 2001). This diverse system consists of species rich plankton communities, seagrass meadows, shellfish bed, diverse fish, and nesting areas for waterfowl (Kennish, 2001). The tidal basin covers 279 square kilometers in surface area and stretches north to south for almost 86 kilometers (Kennish, 2001). This lagoon system is separated from the Atlantic Ocean by a network of barrier islands. The barrier beaches consist of primary and secondary dunes while the barriers facing the bay primarily consist of extensive tidal flats, salt marshes, and upland forests (Kennish, 2001). The bay system is also very shallow, with an average depth of 1.5 meters in the middle of the system and an average depth of 1-3 meters in the southern portion (Kennish, 2001). The mean tidal range is 0.95 meters at Barnegat Inlet and 0.15 meters further into the Bay; thereby this system is microtidal. (FWS, 1995). Due to the shallow depth and minimal tides, wind is the primary driver that influences water circulation patterns (Kennish, 2001). There are currently 10,900 ha of salt marsh in Barnegat Bay. However the bay has been extensively developed during the last century, which has resulted in the loss of approximately 28% of bay's historic salt

marshes due to dredging and infilling (Kennish, 2001). Thus, there is need for the wise and effective management of remaining resources.

Study sites were chosen in areas of Barnegat Bay that have been managed with OMWM practices (Figure 4). Three wetlands, spatially arranged north to south, were

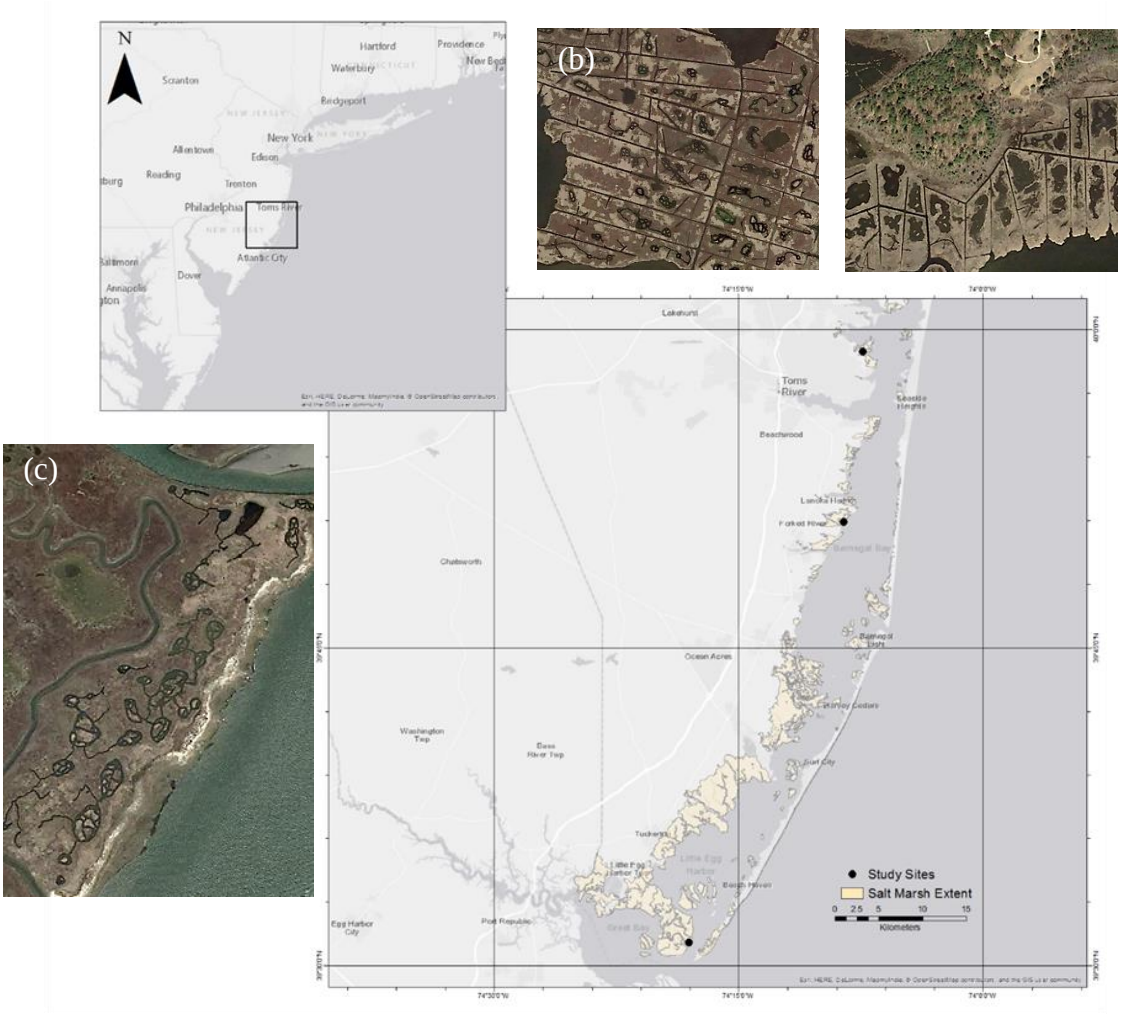

Figure 4. Study sites in Barnegat Bay, Ocean County, NJ. Arranged north to south, Cattus Island (a), Upper Barnegat (b), and Tuckerton (c). Please note that all sites were managed with OMWM.

chosen to compare GHG fluxes: north Barnegat Bay (Cattus Island, 39°58’ 56.8”, -74°07’24.8”), central Barnegat Bay (Upper Barnegat Bay Wildlife Area, 39°50’ 56.5”, -74°08’35.0”), and south Barnegat Bay (JCNERR, on the Tuckerton peninsula,

39°31’15.7”, -74°17’58.4”) (Figure 4). By arranging our study sites from north to south, we intended to capture the north to south gradient in salinity found in Barnegat Bay (Kennish, 2001). The vegetation of these sites consisted of primarily of *Spartina alterniflora* (mixed tall and short form) and *Spartina patens*.

Greenhouse gas (GHG) flux measurements were performed during the 2017 growing season. Monthly measurements were carried out from early May to late August and then every other month starting in September though January. Methane and carbon dioxide exchange were measured at three to five OMWM ponds per site per visit, including measurements for intact marsh vegetation, for bare areas created by sediment sidecasts during pond construction, and from ponds (Figure 5) using floating chambers via an LGR Ultra-Portable Gas Analyzer, hereafter referred as the LGR. For each site per visit, carbon dioxide exchange was also measured at two to four different OMWM ponds using a LI-COR LI-8100 CO_2 flux analyzer.

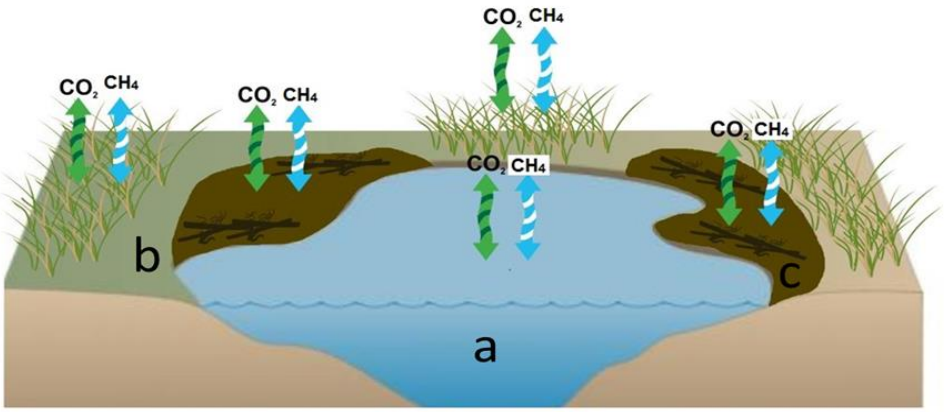

Figure 5. The habitats that are associated with each OMWM pond. Study habitats included a) pond, b) intact marsh, and c) bare ground areas (IAN, 2017).

This equates to measurement of approximately eight ponds per site, per visit with the two analyzers; at half of the sites CO_2 and CH_4 exchange was measured; at half of the sites only CO_2 exchange was measured. GHG measurement sites were generally 200 m

distant from each other. Measurements were made within 3 hours of low tide during daylight hours between 9 AM and 2 PM.

2.2 Gas Exchange Field Methods

The two gas analyzers used for this study were portable infrared (IR) gas analyzers. Since different gases absorb IR light at different wavelengths, IR technology can be used to calculate the concentration of that specific gas based on absorption with a sensitivity that is 500 times better than of a gas chromatograph (GC) (Brannon et al., 2016). The gas analyzers were connected to transparent acrylic static chambers (30.48 cm diameter x 20.32 cm tall) via nylon tubing to create a closed system within which the rate of change of gas concentration over time can be measured. A backflow-prevention valve controlled pressure equilibration within the static chambers. Additionally, a small fan was attached inside the chamber for proper air mixing during the measurements. The design of the static chambers was adapted from Martin et al., 2015. For bare and intact marsh GHG measurements, collars were installed to support the static flux chamber. For the pond measurements, a floating base was used to support the chamber. The terrestrial collars were constructed from PVC rings (30.48 cm diameter x 12.7 cm tall) and the floating base was made from closed cell foam with a hole cut out to hold the static chamber in place. Gas measurements were five minutes per habitat type. GHG fluxes were calculated using the volume of chamber and area of the footprint (Martin et al., 2015). The ideal gas law:

$$(PV = nRT)$$

was used to calculate the gas concentration rate of change over time with the measured air-temperature and atmospheric pressure (Martin et al., 2015). Since the gas measurement incubations were relatively short in time for this study, ebullitive fluxes were not accounted for and thus could represent underestimates in CH_4 emissions (Martin et al., 2015).

At each measurement plot, chamber temperature, and photosynthetic radiance were measured. Temperature and photosynthetic irradiance were monitored within the chamber during each measurement using a Hobo pendant data logger (Onset Computer Corporation, Bourne, MA) and an Odyssey Photosynthetic Active Radiation logger (Dataflow Systems Limited, Christchurch, New Zealand), respectively. It should be noted that the PAR values are uncalibrated; as such the uncalibrated data were used to reveal trends for statistical purposes only. These uncalibrated data are relative to the actual PAR values during each measurement; they are simply missing an offset value from a calibrated source. While we did perform calibration with a PAR logger, we lacked a multi-point calibration as the side-by side comparison occurred for several hours near noon and little variation in PAR was found.

Porewater salinity and soil temperature were taken to serve as additional edaphic variables for analysis. Porewater was extracted using a stainless steel sipper connected to a syringe at depths between 20 to 30 cm. The sipper was inserted into the wetland soil and the syringe was used to draw up porewater from the soil, which was measured using a refractometer calibrated with distilled water prior to use. Temperature measurements were taken 10 cm below the surface. Additionally, vegetation type and percent cover were recorded for the intact marsh gas measurement. Gas fluxes were measured during

light conditions and dark conditions for the bare ground and intact marsh habitats; but only light conditions were measured for the pond habitat due to difficulties with shading off large areas.

Previous work has alternatively advocated for permanently installed or temporarily installed chamber collars (Weston et al., 2014). Permanently installed collars would be less subject to sampling bias due to elevated fluxes from the disturbance of installing the collar. Alternatively, permanently installed collars modify the marsh environment, through trapping of sediment and water, and restrict sampling to the same exact points during each sampling trip. In our study, we installed collars temporarily and allowed 20+ minutes for equilibration. We also conducted a test, where we measured gas exchange using collars installed the same day vs. bases installed two days previously.

2.3 Imagery and Processing

A spatial analysis of Barnegat Bay was conducted to determine the extent of tidal marsh converted to pond by OMWM practices. The spatial analysis was performed using high-resolution aerial imagery in conjunction with object-based analysis (OBIA). High-resolution 2014 aerial photography was used in the OBIA (NOAA, 2014); the imagery contained four bands (near infra-red, red, green and blue); spatial resolution was 0.35 m. Each tile in this dataset was 1km by 1km and 65 tiles were analyzed using OBIA.

In OBIA, images are first segmented into groups of homogenous pixels known as "image-objects." These object images are then classified into categories of interest by unsupervised, supervised, or rule-based algorithms (Dronova, 2015). Using remote sensing techniques to extract information from wetland habitats is a known challenge

because of the heterogeneous nature of the image pixels resulting from variability along the wetland landscape (e.g. soil moisture, vegetation, spectral variables) (Dronova, 2015). OBIA incorporates image object shape and texture as well as the spectral values (i.e. NIR, Red, Green, Blue, image bands) of the input image for classification. The use of OBIA in wetland landscapes is still relatively new, thus, using this technique to determine wetland vegetation loss from OMWM practices is novel. The software used for our OBIA was eCognition Developer ® Version 9.3. The workflow used for the OMWM classification is as follows: segmentation, classification, and manual post editing. A detailed description of the methods for each process will be discussed below.

2.3.1 Segmentation

A good segmentation is the crucial first step for accurate classification. Image segmentation groups homogenous pixels into objects and considers them as spatially independent bodies or "primitive segments" (Moosavi et al., 2013; Zhang et al., 2005). These primitive segments should correspond to real world objects of interest and it can be difficult to create an optimal segmentation output. A combination of two segmentation algorithms produced the best segmentation output.

The first segmentation process used in eCognition was the common multi-resolution segmentation (MRS). MRS is a bottom up, region-merging method that starts with one-pixel objects (Benz et al., 2004). The small image objects are merged into larger objects based on their spectral similarity and shape characteristics. These defining characteristics are weighted parameters that make up the heterogeneity parameter (Gao et al., 2006). When addressing the spectral heterogeneity parameter, the segmentation is

performed using the sum of the standard deviation of the spectral values in each color band of the inputted image while also weighing on certain user-specified layers (i.e. NIR, Red, Green, Blue,) (Gao et al., 2006; Benz et al., 2004). To avoid branched and fractal objects during the segmentation, spectral heterogeneity and spatial heterogeneity criterion are merged to reduce the deviation from a compact or smooth shape (Gao et al., 2006). Prior to the merge of two adjacent objects during the segmentation process, the spectral heterogeneity parameter coupled with the smoothness and compactness heterogeneities needs to be calculated and it is defined as the increase of heterogeneity or scale parameter (Gao et al., 2006). If this number is exceeds the user prescribed, "scale parameter" then no fusion takes place and segmentation stops (Benz et al., 2004). If the scale parameter is increased, more objects can merge therefore growing the object-image. For this study, the scale parameter was adjusted based on trial-and-error practices. There is no concrete standard for the pre-classification assessment for segmentation of wetland landscapes, and they ultimately rely on fuzzy logic approaches. However, the trial-and-error practices are common approaches to segmentation in OBIA (Dronova, 2015).

In addition to the MRS, a spectral difference segmentation was performed on the image object level that resulted from the MRS. This segmentation algorithm merges neighboring image objects according to their mean image layer intensity values (Definiens, 2012). The neighboring image objects are merged if the difference between their layer mean is larger than a certain user prescribed threshold/maximum spectral difference (Definiens, 2012). This segmentation is designed to refine the segmentation results. To determine the appropriate number for the maximum spectral difference, trial-and-error practices were performed to obtain the optimal primitive segments.

Since eCognition can only handle an image less than 5,000 pixels by 5,000 pixels, the segmentation processes needed to be carried out on each Barnegat Bay imagery tile that contained salt marsh landscape. Sixty-five tiles that contained salt marsh were segmented for this study. The scale parameter used for the multiresolution segmentation was 15, and the NIR layer was weighted 2. We used default shape heterogeneity values of 0.5 for compactness and 0.1 for shape. The spectral difference scale parameter was 15 with a weight of five on the NIR. This segmentation process was determined to be optimal to segment individual OMWM areas as image-object primitives.

2.3.2 Classification

Once appropriately segmented, a simple binary OMWM/non-OMWM classification was performed on the image-object primitives by supervised rule-based methods. Out of the red, green, blue, and NIR color bands of the aerial photographs, the NIR band was deemed the most important layer band to separate OMWM from non-OMWM. The image-object primitives have numerous attributes that the user can choose to classify from (e.g. shape, texture, mean RGB, NIR mean values, area, length, etc.).

In this case, mean NIR values and the pixel size of the image-object primitives were the attributes used to discriminate between OMWM and non-OMWM. The mean NIR values of OMWMs ranged from 55-80 in most of the tiles with some discrepancies in some of the aerial tiles in Barnegat Bay. This mean NIR value is significantly lower than wetland vegetation with a mean NIR above 100 thus; vegetation can be easily delineated from open water. As for the delineation of OMWMs from other sources of water (e.g. channels, natural ponds, bay), a combination of mean NIR and object-image

size and shape were used to delineate other sources of water from OMWMs. As previously discussed, a good segmentation is critical for the classification process and the image-object primitives for channels, natural ponds, and the open bay were significantly larger in area than OMWM areas; thus it was utilized as a parameter to delineate OMWMs. The parameters (i.e. mean NIR, and pixel size) to delineate the OMWMs were used to create thresholds to delineate and classify potential OMWMs. The user-defined parameter thresholds (i.e. mean NIR, area) for this binary classification varied with each tile in Barnegat Bay resulting from the spectral variables associated with the aerial image; therefore relied on significant user supervision (Dronova, 2015).

2.3.3 Export and Post Editing

After the classification of the 65 tiles, the OMWM class for each tile was exported to a polygon shapefile with its associated pixel area. Exporting the OMWM classes to shapefiles allowed for post editing of incorrect OMWM classifications in ArcGIS 10.2.2. During the classification process in eCognition, some primitive objects were incorrectly classified as OMWMs. The primitive objects that could be incorrectly classified as OMWMs were parts of channels, natural ponds, shadows of houses, and some other noise associated with the varying spectral values. These incorrect classifications were manually deleted within ArcMap to generate an accurate count and area of the OMWMs in Barnegat Bay.

2.3.4 Accuracy Assessment

To determine the accuracy or performance of the OBIA, a binary confusion matrix was produced. To produce the matrix, 125 points were overlaid on the exported

OMWM polygon shapefiles from the OBIA and another 125 points on a salt marsh polygon obtained from NJDEP. Each point was analyzed and put into one of four categories: 1) OMWMs that were classified correctly; 2) OMWMs that were incorrectly classified as Non-OMWM; 3) Non-OMWMs that were incorrectly classified as OMWM; 4) Non-OMWMs classified correctly. In order to stay objective during this analysis, we defined an OMWM as a round shape pond that was not connected to the channel network and had inner islands within the pond. OMWMs have these distinguishing characteristics that distinguish them from natural ponds.

In addition to determining the classification performance of OBIA, we wanted to determine the accuracy of OBIA when extracting the correct area of each OMWM pond. A simple percent difference was calculated between manual heads up digitization of OMWM and the OMWMs that were classified using OBIA. Fifteen random OMWM ponds were selected and traced in ArcMap using the editor tool bar and the area of each trace were calculated. The associated fifteen random OMWM ponds then were compared to those same OMWM ponds that were classified by eCognition and the percent difference of area was calculated.

2.4 Statistical Analyses

At each site, study plots (consisting of the three habitat types) were measured for GHG flux, porewater salinity, PAR, and soil temperature. These measurements equate to nine data points for each study plot with approximately five to eight study plots per visit, totaling approximately 60 to 96 data points per site per visit. Dark GHG measurements were limited to only the bare ground and intact marsh habitats due to difficulties with

fully shading ponds for measuring ecosystem respiration. The growing season consisted of May, June, August, and September for this study. We also measured fluxes in the fall and winter months (November and January) to determine the effects of seasonality.

The light measurements reflected net ecosystem exchange (NEE) while the dark measurement represented ecosystem respiration. Gross primary production (photosynthesis) can be determined by the difference between NEE and ecosystem respiration. However, for this study we focus primarily on NEE as a means for carbon sequestration potential as it takes in account photosynthesis and ecosystem respiration. We up-scaled NEE and dark measures using day-lengths through the growing season to estimate net fluxes of carbon dioxide across habitats.

Analysis of variance was used to identify statistically significant differences in GHG exchange between habitat types, sites, edaphic variables, and seasonality. To account for deviation in normality of the gas flux data, data were rank transformed prior to ANOVA analyses (Martin et al., 2015). Tukey's HSD, a post-hoc test, was used for pairwise comparisons. Non-parametric regression (Spearman's Correlation Analysis) was used to identify relationships between GHG exchange and PAR, salinity, and temperature. We used a Wilcoxon signed Rank test (a non-parametric repeated measures test) to determine whether there were significant differences in gas exchange between the temporarily and permanently installed chamber collars. Statistically significant differences were determined using $\alpha = 0.05$.

3. Results

3.1 Carbon Dioxide Exchange

Because there were no significant differences in GHG exchange between sites (CO_2 NEE; F= 0.67, p= 0.51; CO_2 ecosystem respiration, F=2.75, p= 0.07; CH_4 light measure, F=0.112, p=0.90; CH_4 dark measure, F= 1.25, p=0.30), we excluded site as a factor from our ANOVA models. Additionally, measures of CH_4 exchange under light and dark conditions did not differ according to a paired Wilcoxon Signed Ranked Test (W=7358, p=0.44). Therefore, we used only light CH_4 measurements in our ANOVA models.

We found that during the studied months, photosynthetic uptake of carbon dioxide occurred during all months for the intact marsh and bare ground habitats, except for small emissions in January. Photosynthetic uptake between the two habitat types were significantly different (p<0.01) (Figure 6). The average photosynthetic uptake in the intact marsh habitat was -8.4 ± 0.6 µmols m^{-2} s^{-1}. The average photosynthetic uptake for the bare ground habitat was -0.68 ± 0.2 µmols m^{-2} s^{-1}.

Seasonality (i.e. month) significantly affected the photosynthetic uptake within the intact marsh habitat ($F_{5,104}$ =16.94, p<0.001) but did not affect the photosynthetic uptake of the bare ground habitat ($F_{5,104}$ = 4.32, p= 0.1). Post-hoc test results indicate significant differences between many month pairings (Table 1).

Ecosystem respiration was occurring in both intact marsh and bare ground areas (Figure 7). The average ecosystem respiration rate of the intact marsh habitat for all studied months was 6.6 ± 0.5 µmols m^{-2} s^{-1} while the bare ground respired at an average

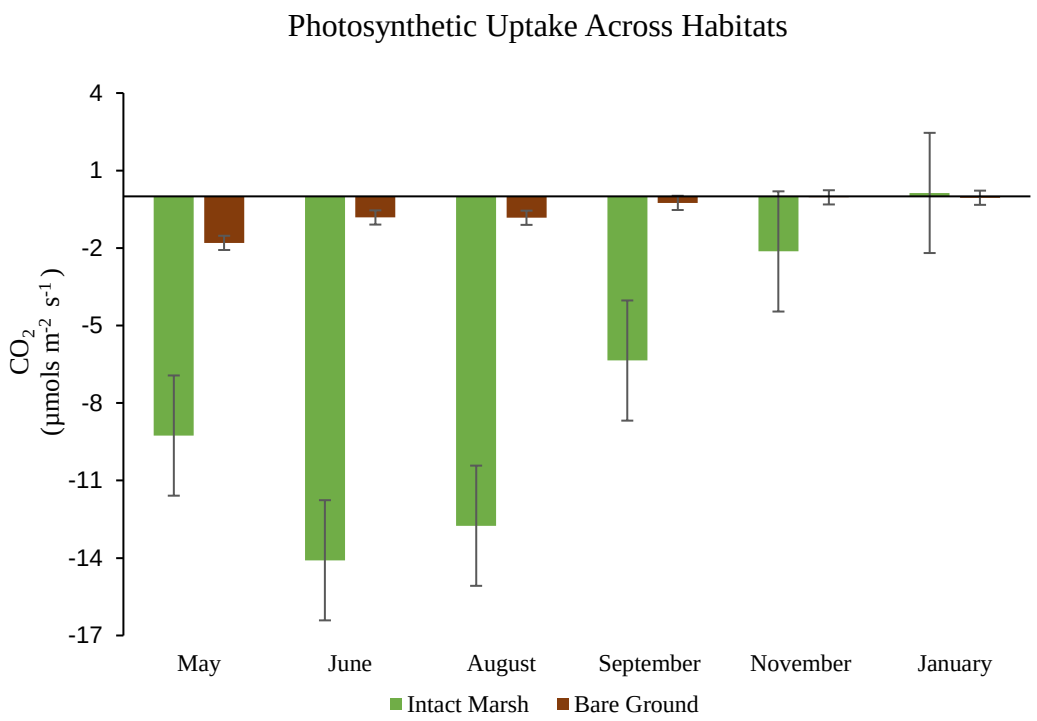

Figure 6. Photosynthetic uptake between intact marsh and bare ground habitats during the studied months. Note the seasonal pattern of photosynthetic uptake during May through November until small emissions arise in January. Error bars reflect standard error for the ecosystem respiration for all sites for each month.

Table 1. Results of Tukey's post-hoc test, identifying which months were significant different for the habitats that experienced a significant seasonal effect.

Intact Marsh PS	**Intact Marsh R_{eco}**	**Intact Marsh NEE**	**Bare ground R_{eco}**	**Pond NEE**
Jan - Aug, p = 0.005	Jan - Aug, p< 0.001	Nov - June, p= 0.058	Nov - Aug, p = 0.02	May - Aug, p = 0.02
Nov - Aug, p<0.001	Nov - Aug, p<0.001		Nov - June, p< 0.001	May - Jan, p< 0.001
June - Jan, p<0.001	June - Jan, p<0.001			Nov - May, p = 0.04
Nov - Jan, p = 0.02	May - Jan, p = 0.005			Sept - May, p< 0.001
Nov - June, p<0.001	Sept - Jan, p = 0.04			Sept - Nov, p = 0.04
Sept - June, p<0.001	Nov - June, p<0.001			
Nov - May, p =0.002	Sept - June, p<0.001			
	Nov - May, p =0.002			

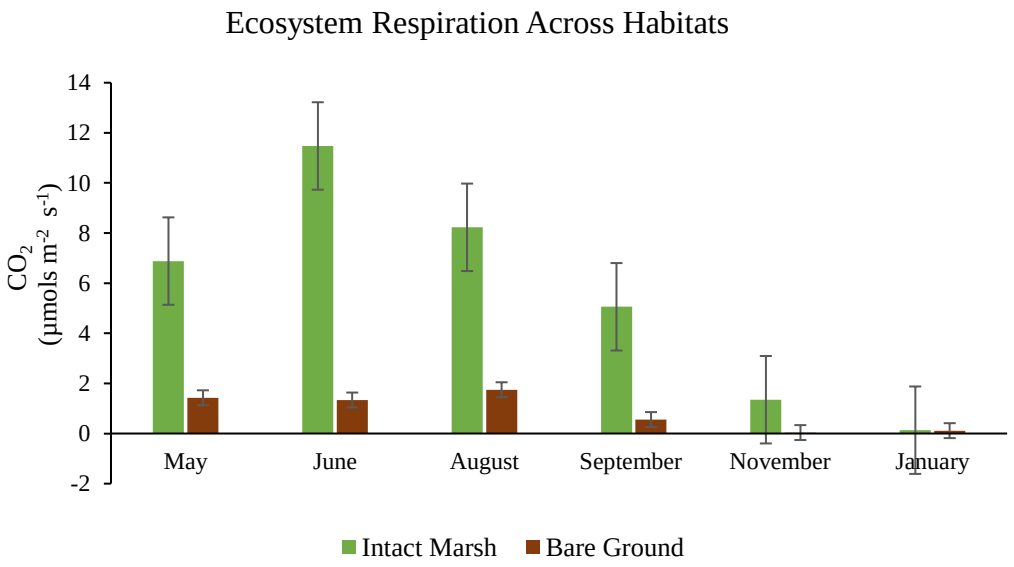

Figure 7. CO_2 ecosystem respiration of carbon dioxide from the intact marsh and bare ground areas. As seen from the graph, ecosystem respiration is positive for all studied months for the two habitats and peaked in June when temperatures were the warmest. The ecosystem respiration trend declined as the growing season ended. Error bars reflect standard error for the ecosystem respiration for all sites for each month.

rate of 1.0 ± 0.2 µmols m^{-2} s^{-1} for all studied months. Ecosystem respiration between the two habitat types were significantly different ($p<0.01$) (Figure 7). Seasonality significantly affected the CO_2 ecosystem respiration within the intact marsh habitat ($F_{5,104}$ =14.29, $p<0.001$) as well as affected the ecosystem respiration of the bare ground habitat ($F_{5,104}$ = 4.32, p=0.001). Ecosystem respiration from the intact marsh habitat in June was almost 90 times higher than the ecosystem respiration that we saw in January. Significant differences were found between many month-pairs using post-hoc tests (Table 1).

We found that during the studied months, NEE of CO_2 was generally negative (i.e, carbon dioxide taken up in the intact marsh habitat) (Figure 8). The average CO_2 NEE for the intact marsh habitat for all studied months was -1.7 ± 0.3 µmols m^{-2} s^{-1}. In contrast, the other habitats generally were net emitters of CO_2 (Figure 6). The average CO_2 NEE over the studied months for the bare ground and pond habitats was 0.32 ± 0.1

µmols m^{-2} s^{-1} and 0.23 ± 0.06 µmols m^{-2} s^{-1}, respectively. The CO_2 NEE fluxes were significantly affected by habitat type ($F_{2, 317}$ =28.795, p<0.001). Intact marsh CO_2 NEE

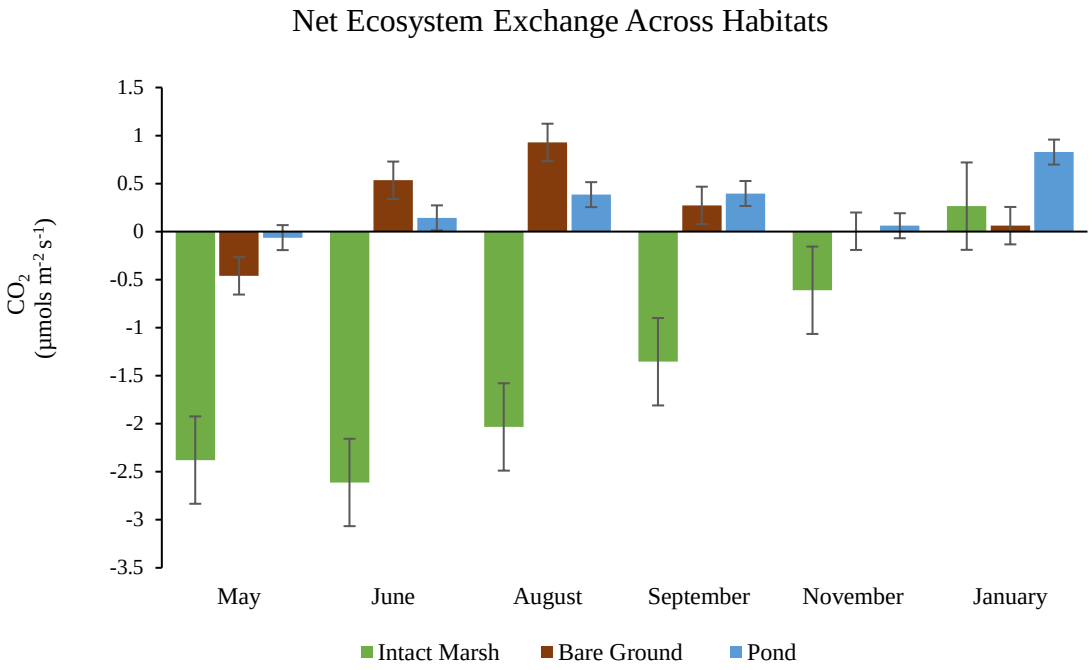

Figure 8. Net ecosystem exchange of CO_2 from the intact marsh, bare ground, and pond areas.

was greatest (p< 0.001), and was significantly different from the bare ground and pond habitats; however, the CO_2 NEE from the pond and bare ground habitats were not statistically significantly different from each other.

There was a clear seasonal pattern of the CO_2 uptake during the studied months, specifically in the intact marsh habitat. Seasonality (i.e. month) significantly affected the CO_2 NEE within the intact marsh habitat ($F_{5, 104}$ =2.54, p= 0.03). Seasonality also significantly affected the CO_2 NEE in the pond habitat ($F_{5, 104}$ =7.385, p< 0.001). However, seasonality did not have an effect on the CO_2 NEE within the bare ground

habitat ($F_{5, 104}$= 0.17, p= 0.97). Post-hoc test results (Table 1) identify what months were significantly different for the three different habitats.

3.2 Methane Exchange

Habitat type had a significant effect on CH_4 fluxes ($F_{2, 315}$ =0.894, p< 0.001). There were no significant differences between the CH_4 fluxes in the pond and intact marsh habitats; however, the marsh and pond CH_4 flux were significantly higher than the bare ground habitat. Although, variable, the CH_4 exchange significantly differed throughout the studied months ($F_{5, 315}$ = 21.84, p<0.001) when all habitat types were considered. There was no interaction effect between month and habitat type ($F_{10, 310}$ = 1.224, p= 0.254). The intact marsh and bare ground habitats had the highest CH_4 fluxes recorded in September while the pond habitat experienced the highest fluxes in January (Figure 9).

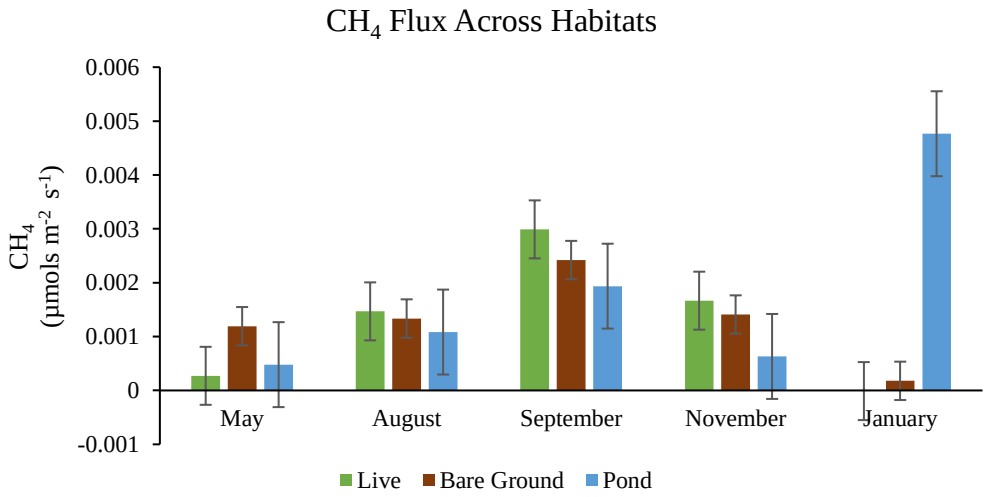

Figure 9. Methane fluxes from the studied months. Please note that all habitats produced variable rates of methane; however, all measurements were positive. Error bars reflect standard error for the methane exchange for all sites for each month.

3.3 Environmental Variables

Over the studied months, the measured environmental variables significantly changed; including soil/pond temperature ($F_{5, 263}$ = 150.2, p< 0.001), PAR ($F_{5, 57}$ = 10.27, p< 0.001), and porewater salinity ($F_{5, 299}$ = 3.511, p= 0.004). Average porewater salinity across all habitats significantly differed between sites ($F_{2, 299}$ =169, p<0.001). The average porewater salinity was 20 ± 0.4 for Cattus Island, 26 ± 0.5 for Upper Barnegat Bay and 35 ± 0.8 for Tuckerton, all categorically polyhaline.

Because our general linear models identified interaction effects between soil temperature and gas exchange between habitats, we used non-parametric regression to examine the relationships separately. Of the three habitats, bare ground had the warmest recorded temperatures.

Soil temperature in the intact marsh was negatively correlated with photosynthetic uptake (*Spearman's r* = -0.53, p< 0.01). (Note that the greatest CO_2 uptake was identified with a negative sign, thereby making the correlations negative.) Therefore, photosynthetic uptake increased with increased temperature. Soil temperature in the intact marsh habitat was positively correlated with CO_2 ecosystem respiration (*Spearman's* r =0.65, p<0.001) while no significant correlation was found between CO_2 NEE and soil temperature (*Spearman's r*= -0.20, p=0.06).

There was no correlation between soil temperature and photosynthetic uptake in the bare ground habitat. However, there was a positive correlation between soil temperature and CO_2 NEE (*Spearman's r* = 0.29, p=0.006) as well as CO_2 ecosystem respiration (*Spearman's r*= 0.36, p =0.008) in the bare ground habitat. (Note that the smallest CO_2 uptake was identified with a positive sign; therefore, with increased

temperature there is a decrease in CO_2 uptake.) No correlation between CO_2 NEE and pond temperature was found in the pond habitat (*Spearman's r*= 0.17, *p* =0.2). There was no correlations between CH_4 exchange and soil/pond temperatures within all studied habitats.

There were significant differences in porewater salinity and habitat type ($F_{2, 299}$ = 10.35, $p< 0.01$). The bare ground habitat had the highest porewater salinity (29 ± 0.9) and was significantly different from the pond habitat ($p< 0.001$) but not significantly different between the intact marsh habitat (p=0.3). Additionally, porewater salinity in the intact marsh was significantly higher than the pond habitat (p= 0.01).

Porewater salinity did not have an effect on photosynthetic uptake in the intact marsh (p=0.1) or bare ground habitats (p=0.2). Additionally, there was no effect of porewater salinity on CO_2 NEE in all habitats ($p > 0.05$). In terms of CO_2 ecosystem respiration, there was a significant effect by porewater salinity when the two habitats were included. However, there was an interactive effect between porewater salinity within the bare ground habitat (p=0.004) only. Therefore, we speculated that the ecosystem respiration from the bare ground habitat was driving the model for the effect of porewater salinity. When individual bare ground salinity was analyzed separately, it had a positive correlation with CO_2 ecosystem respiration (*Spearman's r*=0.26, p=0.007). There was no significant effect by porewater salinity on CO_2 ecosystem respiration in the intact marsh habitat (p=0.6). There was a positive correlation between porewater salinity and CH_4 exchange (*Spearman's r* = 0.48, $p<0.001$) in the intact marsh habitat while there was no effect of porewater salinity on the CH_4 exchange flux in the pond or bare ground habitats. ($p>0.05$).

Table 2. Correlations between the GHG fluxes and the measured environmental variables. Only significant spearman correlations are reported in the table.

	Photosynthesis	Respiration $_{Eco}$	NEE	CH_4
Soil Temperature	**Intact Marsh** *Spearman's r = -0.53, p<0.001**	**Intact Marsh** *Spearman's r=0.65, p<0.001** **Bare ground** *Spearman's r = 0.36, p<0.001**	**Intact Marsh** *Spearman's r= -0.20, p=0.06* **Bare ground** *Spearman's r =0.29, p=0.006**	
PAR	**Intact Marsh** *Spearman's r = -0.50, p<0.01**		**Intact Marsh** Spearman's r= -0.32, p=0.01*	
Porewater Salinity		**Bare ground** *Spearman's r =0.26, p<0.001**		**Intact Marsh** *Spearman's r= 0.49, p<0.001**

PAR was negatively correlated with CO_2 uptake (photosynthesis) (*Spearman's r* = -0.50, *p*<0.01) and CO_2 NEE (*Spearman's* r = -0.32, p=0.01) in the intact marsh habitat. Please note that the greatest CO_2 uptake was identified with a negative sign, thereby making the correlations negative. Therefore, with increased PAR, there is an increase in photosynthetic uptake of carbon dioxide. There was no significant correlation found between PAR and CO_2 NEE or photosynthetic uptake within the bare ground areas. Additionally, we found no significant correlation between PAR and CO_2 NEE within the pond habitat.

3.4 Collar Test

Based on the Wilcoxon Signed Ranked Test, we determined that there was no statistically significant difference in the CO_2 NEE (*V*= 11, *p*= 1.0) and ecosystem respiration (*V*=13, *p*=0.69) during collars inserted the same day vs. several days earlier,

indicating that the disturbance due to the collar insertion did not influence measured GHG exchange. We were unable to measure CH_4 for this test.

3.5 Potential Drivers of GHG Flux

In the intact marsh habitat, photosynthetic uptake was a function of temperature and PAR, while ecosystem respiration was a function of temperature, and CO_2 NEE was dependent on PAR. In the bare ground habitat, CO_2 NEE was found to be temperature dependent, while ecosystem respiration was dependent on temperature and salinity. The pond habitat had variable CO_2 NEE fluxes throughout the studied months; we found no significant correlation with the measured environmental variables. CH_4 emissions from were also variable. Only the intact marsh habitat experienced a positive relationship for CH_4 with porewater salinity and no other correlations were determined for the other habitats with other the measured environmental variables.

3.6 Spatial Analysis

The results of the OBIA show that extensive marsh area has been disturbed by OMWM practices (Figure 10). The total area directly impacted by OMWM practices was approximately 3,077,000 square meters (307.7 ha). The spatial extent of salt marshes is 1.09×10^8 square meters (10,900 ha), not accounting for the destruction associated with OMWM practices. Therefore, when accounting for area lost due to OMWM practices, it equates to almost 3% of the total area of salt marsh in Barnegat Bay (Table 3). We used the aggregate function in ArcGIS to aggregate ponds <5m away to estimate the number

and average size of ponds constructed for mosquito control: the number of ponds summed to 7,030 ± 210; while mean pond area was estimated as 440 ± 23 m^2.

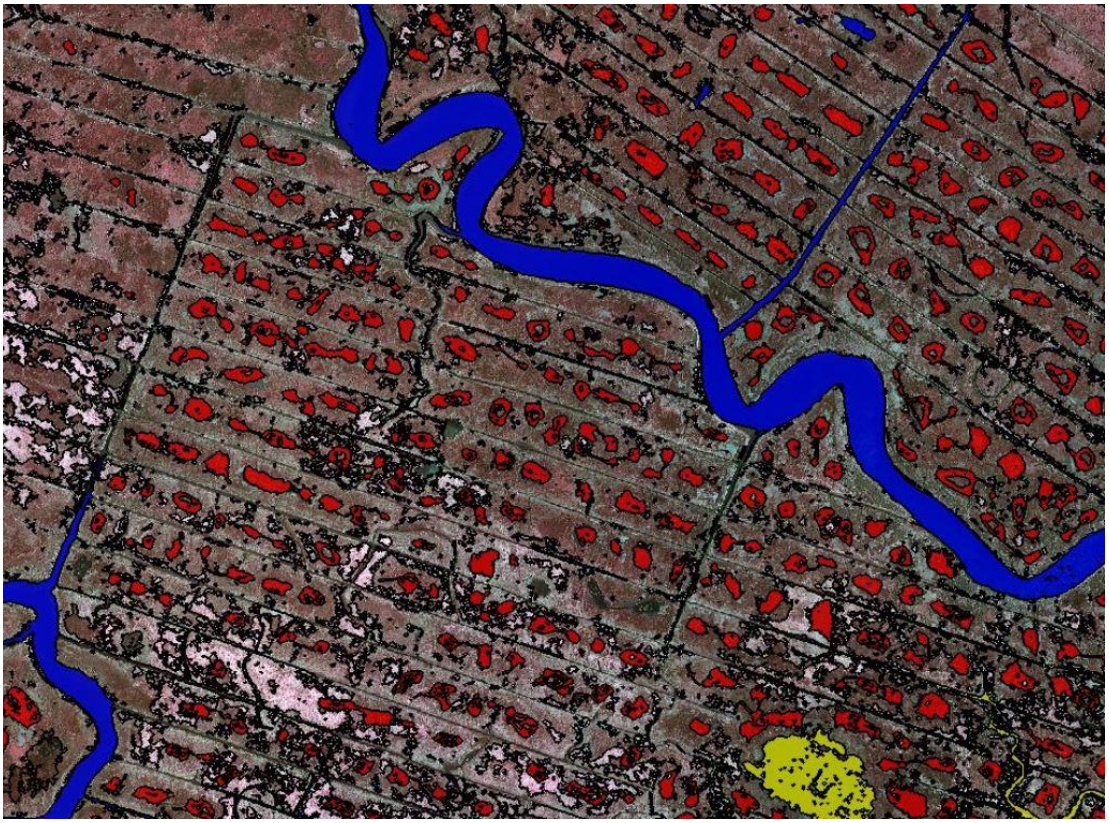

Figure 10. Results of output of the OBIA of a portion of Barnegat Township where extensive OMWM has taken place. Note that the OMWM ponds are in red and the channels were not included as OMWM.

Table 3. The average area lost due to OMWM practices in Barnegat Bay, NJ. Error associated with the values were determined using a simple error analysis that determined the classified area accuracy.

Average OMWM Area	440 ± 23 m^2
Total Salt Marsh in BB	1.09 x 10^8 m^2
Total Area Affect by OMWM	3,080,000 ± 123,000 m^2
% lost due to OMWM	2.82

Based on expert classification of random points, our OBIA classification was 96.5% accurate (Table 4). Errors of commission and errors of omission were symmetrical, indicating no systematic under or over prediction. Additionally, the percent difference between manual and automated classification was 5%.

Table 4. Binary confusion matrix for the classification of OMWMs and NON-OMWMs. The user accuracy is the percent accuracy with commission error subtracted while producer accuracy is the percent accuracy with omission error subtracted. *Kappa* = 0.928.

	OMWM Present	**OWWM Classified Absent**	**Users Accuracy**
OMWM Classified Present	126	4	97%
OMWM Classified Absent	5	115	96%
Producers Accuracy	96%	97%	96.5%

4. Discussion

4.1 Impacts of OMWM on Greenhouse Gas Exchange

In this study, our goal was to determine the areal extent of Barnegat Bay coastal wetlands that had been converted to open water ponds for mosquito control, as well how this has shifted the balance of greenhouse gas exchange in Barnegat Bay coastal wetlands. Here, we compare the disturbance posed by OMWM with other disturbances present in coastal wetlands, both in terms of extent and magnitude. We discuss the accuracy of our image classification technique relative to other methods of coastal habitat classification. We then focus on greenhouse gas exchange and implications of OMWM construction on carbon sequestration. We conclude with a discussion of management implications of this work.

Coastal marshes have been subject to historic land use conversion for row crop agriculture, pasture land, aquaculture, residential, and industrial development. Their simple topography and coastal access has made them good candidates for land use conversion. Fertile wetland soils make these ecosystems attractive for farming and grazing, and many of the worlds notable agricultural areas are coincident with former coastal wetlands, such as the Sacramento Delta in California, the coastal plain of the

Netherlands, and the Sanjang Plain in China (Liu et al., 2004; Verhoeven and Setter 2009). Construction of shrimp ponds for aquaculture has disturbed many of the world's mangroves (Naylor et al. 2004). Where land for coastal development has been at a premium (such as the urban areas of Boston, the San Francisco Bay Area, and Washington DC), much of the urban development lies on former coastal wetlands and up to 60-90% wetlands have been lost (Goals Project, 1999; Gedan et al., 2009). In southern New Jersey, disturbance of coastal wetlands began when subsistence farming shifted to a market-oriented system and salt-hay farming proved profitable (Sebold 1998; Gedan et al., 2009). Salt hay farming relies on high marsh species, particularly *Spartina patens*, and to increase acreage of this species, the customary practice was to drain the marsh by ditching (Tonjes 2013). Ditching for salt hay farming continued into the 1930s with the addition of grid ditching for mosquito control.

In this study we found that 3% of the wetlands in Barnegat Bay have been directly disturbed by the construction of ponds for mosquito control, creating over 7,000 anthropogenically excavated pools in Barnegat Bay. One potential analog for considering this disturbance is the construction of shrimp ponds in coastal wetland for aquaculture. In comparison with the water management areas in New Jersey, shrimp farm ponds tended to be larger, more spatially aggregated, constructed more rapidly, and associated with negative environmental impacts such as eutrophication (Dahdough-Guebas, et al. 2002; Muttitanon and Tripathi 2005; Verhoeven and Setter 2009). In comparison with shrimp ponds, the disturbance posed by water management areas in Barnegat Bay has been mild. However, this practice was put in place at least in part as a habitat enhancement technique. How then do the excavated ponds compare with natural

ponds found in coastal wetlands that have been eliminated by past drainage practices? In a study of salt marsh pools in the Northeast from Long Island to Maine, Adamowicz and Roman (2005) found that natural pools in unditched marshes covered an average area of ~200±100 m^2, at a density of 13±7 pools per hectare and comprised 9±4% of habitat cover. Although this study focused on New England where marsh pools are possibly more common as they include kettle ponds in glacially deposited outwash sediments (Sorrie 1994), our study identified some differences between anthropogenic ponds and the natural ponds described by Adamowicz and Roman (2005). The Barnegat Bay anthropogenically constructed ponds tended to be larger, more homogenously sized, and fewer in number than the natural ponds, which may alter the balance of habitat provision. And although the overall area of pond construction summed to only 3% of Barnegat Bay's wetlands, in the areas where ponds were constructed, we estimate that they covered up to 30% of habitat, far larger than the 9% estimated for natural ponds (Adamowicz and Roman 20045.

Application of OBIA resulted in a highly accurate classification of water management areas (~95% accurate). Gao et al. (2008) concluded that OBIA has an advantage over pixel-based approaches with high-resolution imagery as pixel-based classifiers are confused by the spectral variability in high-resolution imagery. Additionally, previous studies have documented greater accuracy in delineation of wetland classes using OBIA than pixel-based classification methods (Dronova 2015). For example, Myint et al. (2008) found 31% greater accuracy with OBIA than with pixel-based classifiers, and a review of other studies has suggested improvements in accuracy ranging from 7% to 28% (Dronova 2015). However, interpreting the accuracy in such

comparisons can be problematic as OBIA is still a relatively new classification technique and different studies rely on different methods to estimate the final map accuracy (Dronova 2015). Based on a side by side comparison of OBIA and maximum likelihood pixel-based classification on a subset tiles, we found that OBIA classification accuracy was 96% accurate vs. 79% for the pixel-based classification method (Watson et al., 2018). Accurate classification of the extent of the OMWM ponds was crucial as the total area and number of ponds has not been previously documented.

Here, we report on the alteration of a crucial ecosystem service – carbon sequestration – by anthropogenic manipulation of salt marshes in Barnegat Bay, NJ. Through carbon sequestration, salt marshes mitigate emissions from fossil fuel combustion by reducing levels of GHGs in the atmosphere (Mcleod et al., 2011). These ecosystems assimilate carbon via photosynthesis and sequester it long-term due to low rates of carbon remineralization in carbon-dense soils. The clearing of aboveground biomass during the conversion to from intact marsh to open water, reduces carbon sequestration and increases carbon emission (Lovelock et al., 2011).

Our gas flux analysis for the three habitats suggests that the intact marsh habitat is a net sink for CO_2 while the bare ground and pond are net CO_2 sources during our incubation times (Figure 8). During the growing season, the rate of NEE in the intact marsh habitat was similar to other studies measuring CO_2 fluxes from *Spartina alterniflora* (Martin et al., 2015, Moseman-Valtierra et al., 2016). As the growing season ended, the trend of uptake decreased until November and then we saw small emissions from the intact marsh during winter. The bare ground habitats exhibited small emissions throughout the growing seasons, with magnitudes similar to those observed previously

from unvegetated marsh (Martin et al., 2015; Moseman-Valtierra et al., 2016). In bare areas, we saw small rates of photosynthetic uptake presumably due to the presence of photosynthetic microalgae, but uptake values were overweighed by respiratory emissions. From ponds, we saw small but variable carbon dioxide exchange values through the growing season, suggesting that ponds are currently not a strong GHG source or sink. Because gas exchange from wetland ponds has rarely been measured (Moseman-Valtierra et al., 2016), and no relationships were found with environmental variables, it is difficult to attribute GHG exchange in ponds to specific factors. Previous studies focusing on GHG emissions in tidal channels however have reported gas exchange rates as a function of tidal pumping and dissolved carbon dynamics (Cali et al. 2015).

Construction of ponds for mosquito control shifted the wetlands from being a greenhouse gas sink to a greenhouse gas source, both within the footprint of the constructed pond as well as within the area where deposition of sediment has prevented the recolonization of marsh vegetation. We suspect that the CO_2 efflux in disturbed soils in coastal ecosystems is caused by organic matter remineralization by microbes (Lovelock et al., 2011). Lovelock et al. (2011) found an immediate increase of CO_2 efflux after disturbance of mangrove soils, and previous work has suggested that large increases in carbon mineralization are resulting from wetland disturbance (Hopkinson et al. 2012). Bare ground areas surrounding water management areas of Barnegat Bay have persisted in some cases since the ponds were constructed (~40 years), and we were not able to sample at recently constructed ponds. Throughout the study months, we see these bare ground areas negatively contribute to CO_2 balance of the salt marsh.

Within vegetated habitats, we saw assimilation of carbon dioxide across the growing season. The dominant plant species in our study system is *Spartina alterniflora,* a C4 grass that is highly responsive to temperature (Shae 1977). The optimum temperature for photosynthesis for C4 plants ranges from 30 to 40°C with a steep decline when temperatures fall below 15°C (Hofstra and Hesketh, 1969), and this responsiveness to temperature is reflected in our data (Figures 3, 5).

Consistent with previous studies (Martin et al., 2015; Moseman-Valtierra et al., 2016; Bartlett et al., 1987), we saw limited exchange of methane in our sampling of pond, bare areas, and vegetation, with slightly higher methane emission rates in marsh vegetation. Coastal marshes have limited methane emissions due to the substrate competition of sulfate reducers rather than aquatic methanogens (Bartlett et al., 1987, Poffenbarger et al., 2011). Porewater salinity of our three sites was polyhaline and ranged from 19 to 34 ppt. Therefore, the salinity of our measurement sites were still above what Bartlett et al. (1987) and Poffenbarger et al. (2011) have identified as the salinity threshold at which we would expect to see higher methane fluxes. The positive correlation between porewater salinity and methane emissions in the intact marsh habitat is not what we would expect to see and does not agree with other studies measuring from the same species composition (Moseman-Valtierra et al., 2016; Martin et al., 2015). However, an *in situ* study by Chmura et al. (2011) found higher methane emissions from one of two sites with higher porewater salinity but found that the water table was lower for one of the sites. This difference in water table could explain the trend since lower water table increases aerobic conditions, which in turn causes an increase in methane

oxidation (Chmura et al., 2011). We did not measure water table in our study; however, this could explain our GHG correlation with porewater salinity.

Our data suggest slightly higher rates of methane emissions from vegetation. Because aerenchyma (air spaces in plant rhizomes and stems) can transport methane from deeper soil levels, this pathway has been traditionally attributed for 80-90% of the methane emissions from wetlands (Cheng et al., 2007). However, estimates from methane emissions from salt marsh plants are highly variable and abiotic controls could exert stronger influences (Moseman-Valtierra et a., 2016; Nuttler et al., 1988). We did see methane emissions from the pond habitat in winter, although we feel this may be attributable to the disturbance of the ice by the incoming tide, rapid warming conditions through the day, as well as the disturbance we enacted by placing the collar in the frozen ground and through a thin veneer of ice; and may not be representative. Our measurement likely was the ebullitive gas exchange, which is usually seen during spring melt (Huttunen et al., 2003; Call et al., 2015). Thus, our winter pond measurements fluxes could be an overestimation of methane fluxes.

4.2 Carbon Accumulation Rate

When considering the total area of intact marsh converted to open water (Table 2), there was a negative shift in carbon sequestration based on fact that the ponds do not contribute to carbon sequestration (Figure 5).

When scaled to the proper amount of daylight/nighttime hours and days throughout one growing season, the carbon accumulation rate for the intact marsh and bare ground habitat was 480±6.9 g C m^{-2} y^{-1}, 38±2.3 g C m^{-2} y^{-1}, respectively (Figure 10).

However, the pond habitat contributes 7.6± 0.5 g C m^{-2} y^{-1} to the atmosphere (Figure 11). This extrapolation is based our average photosynthetic uptake and respiration in the intact marsh and bare ground habitats and scaled to the amount of daylight/nighttime hours during the growing season (90 days). The pond habitat was scaled to the average CO_2 NEE during the growing season. We show, from this extrapolation, that the intact marsh habitat accumulates carbon at a considerable rate while the carbon accumulation rate of the bare ground habitat is an order of magnitude lower, and the pond habitat contributes small amounts to the carbon budget during the growing season. These growing season accumulation rates relatively agree with the literature (Chmura et al., 2015; Forbich et al., 2015; Forbich et al., 2017). However, when considering the carbon losses from the marsh during the winter months, Forbich et al. (2017) saw rates of about 180 g C m^{-2} y^{-1} for marshes dominated by *Spartina patens*, *Disticilis spicata*, and *Spartina alterniflora*. Therefore, if we compare those values to our findings, we see the annual carbon accumulation to be approximately 300 g C m^{-2} y^{-1}. However, this value can only account for the intact marsh habitat. There are few studies studying bare ground and pond habitats in terms of annual carbon accumulation. Spivak et al. (2017) found that natural ponds neither import or export organic matter, agreeing with our findings. Therefore, there is a need for longer term measurements of GHG fluxes in ponds as well as bare ground habitats because it is still unknown as to how they affect the annual carbon budget of the marsh system.

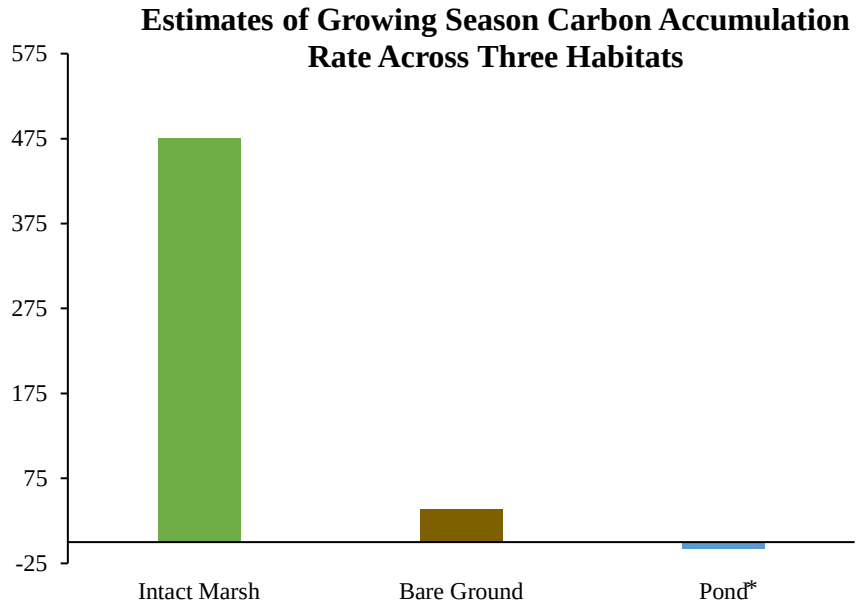

Figure 11. Scaled yearly carbon accumulation rates within the three different habitats during the growing season. *Indicates that the pond habitat extrapolation was scaled on only the NEE values for the growing season.

4.3 Management Implications

Publications focusing on the impact of OMWM to non-targeted wetland resources are rare (James-Piri et al. 2012), and this study provides evidence of how OMWM has shifted the carbon balance of a salt marsh in Barnegat Bay, NJ. Such studies are strongly needed to develop wise management strategies for coastal wetlands.

Previous work focusing on the impacts of OMWM to wildlife habitat, mosquito control, wetland sedimentation, and denitrification have suggested mixed effects of OMWM practices on mosquito populations (James-Pirri et al., 2009; Rochlin et al., 2009), no shift in the highly variable denitrification rates (Velinsky et al. 2017), alterations to sediment transport pathways (Quirk et al. 2015), and little use of ponds by waterfowl (Erwin et al. 1994). While OMWM can clearly reduce mosquito larval density (Rochlin et al., 2012), clear differences between in mosquito production in a BACI-design OMWM monitoring project in Barnegat Bay were obscured by high variability or

possibly the prevalence of mosquito larvicide usage (James-Pirri et al., 2009). One aspect of pond construction that concerned us was the small number of fish we observed in ponds during field sampling, and the lack of connection to the tidal channel network via radial ditches, which would allow the fish to better access the ponds. Without radial ditches and high enough tides, larvivorous fish cannot access these ponds. These observations have also been supported by previous work, which found decreases in killifish populations and an increase of Palaemonidae shrimp populations in water management areas (James-Pirri et al., 2011). Our work suggests that disruption of carbon storage through pond construction has had a negative impact on the carbon storage potential of the wetland, within the area disturbed by pond construction, as well as in the footprint of area disturbed by sediment sidecasts. Taken collectively, these studies raise important questions about the value of OMWM ponds for mosquito control and suggest that the practice reduces the value of wetlands for other purposes, such as carbon sequestration and wildlife habitat. Delaware has already halted the implementation of OMWM, and we urge New Jersey mosquito control agencies to reconsider their practices without further study.

More broadly, our work has also addressed how wetland fragmentation or the replacement of marsh vegetation with open water – which occurs due to coastal development as well as due to sea level rise – is impacting the net carbon sequestration of salt marshes. Our work has suggests that to have accurate measurements of the carbon stocks in our salt marshes for climate change mitigation, we first need have an accurate idea of the actual spatial extent of productive plants because as our data shows, open water does not contribute to GHG sequestration. As policy moves toward expanding

coastal wetlands to carbon markets, we first need to determine the *actual* area of productive plants – as they are the link to mitigate against anthropogenic carbon emissions and the enhancement of climate change (Moseman-Valtierra et al., 2016).

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